

# Vital rates, limiting factors and monitoring methods for moose in Montana



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*Note: All results should be considered preliminary and subject to change; please contact the authors before citing or referencing these data.*

## **Location**

Moose vital rate research is focused primarily within Beaverhead, Lincoln, Pondera, and Teton counties, Montana. Other portions of monitoring (e.g., genetic and parasite sampling) involve sampling moose from across their statewide distribution.

## **Study Objectives (Year 2 of 10-year study)**

For the 2013-2014 field season of this moose study, the primary objectives were;

- 1) Continue to evaluate moose monitoring data and techniques.
- 2) Monitor vital rates and limiting factors of moose in three study areas.

## **Objective #1: *Moose monitoring methods***

### **1.1. Calibrating existing moose monitoring data**

#### *1.1.1. Background*

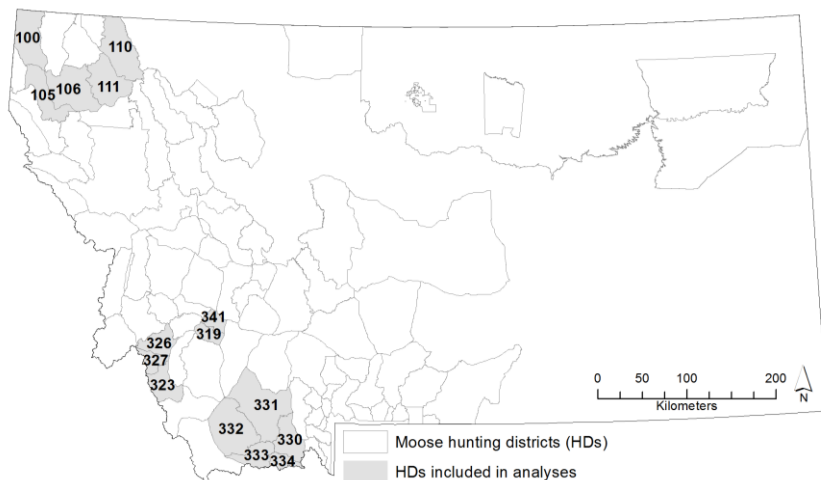
Monitoring of moose and other ungulate populations by Montana Fish, Wildlife & Parks (MFWP) biologists is conducted through a combination of annual aerial survey counts, aerial age-sex composition ratios and hunter harvest statistics from phone surveys. Generally aerial counts require additional information concerning sightability (proportion of animals in the population seen during a survey) before they can be adjusted to statistical estimates of population size. However, time series of unadjusted minimum count data can still allow unbiased estimates of population trend as long as the mean sightability remains constant over time (Harris 1986, Eberhardt and Simmons 1992, Humbert et al. 2009). While aerial surveys often represent the ‘gold standard’ for monitoring moose populations, such methods are costly, and in some scenarios, hunter statistics may provide a cost-effective means of monitoring population trends (Ericsson and Wallin 1999, Bontaities et al. 2000, Ueno et al. 2014).

Here we present some preliminary analyses using aerial count data and harvest statistics for moose populations in 16 hunting districts of Montana to assess: 1) count-based population trends, 2) drivers of uncertainty about those trends, and 3) the shape and strength of relationship between KPUE and aerial counts. These analyses are yet preliminary, and we are still compiling some additional years and forms of data.

#### *1.1.2. Population trends from aerial counts*

We compiled moose monitoring data spanning the years of 1985–2013 for 16 hunting districts in Montana, including the Purcell Mountains (HD 100), Cabinet Mountains (HD 105), Salish Mountains (HDs 106 and 111), Whitefish Range (HD 110), Fleecer Mountains (HDs 319 and 341), upper Big Hole Valley (HDs 323, 325, 326, and 327), Gravelly Range (HDs 330), Ruby and Snowcrest Ranges (HD 331), Blacktail Mountains (HD 332), Centennial Valley (HDs 333 and 334; Figure 1).

Sightability of animals during these flights is a complex product of 3 nested probabilities of: 1) an animal being present within the survey area, 2) an animal being available to be seen, and 3) a given observer actually seeing it (Peters et al. In press). Rather than using sightability models to adjust counts into population estimates, the minimum count approach instead assumes that the net visibility varies about some constant mean. As such, imperfect sightability induces a great deal of observation error, or sampling variance, into minimum count data, which in turn decreases statistical power to detect trends in abundance.



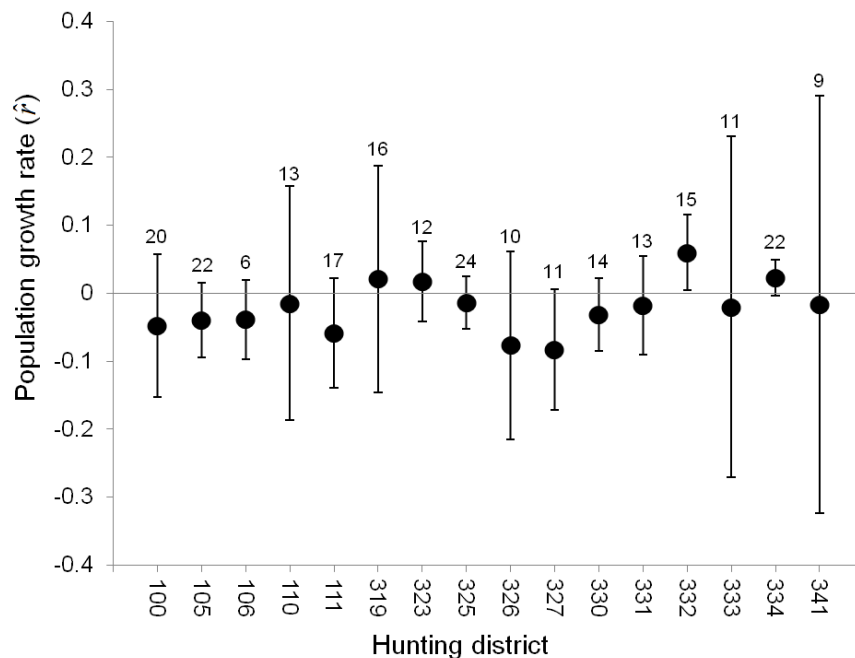
**Figure 1.** Moose hunting districts in Montana and the subset of HDs for which both aerial survey and harvest data were collected during 1985–2013.

Conventional models of population trends using log-linear regression of counts against time have been shown to consistently underestimate the variance of trend estimates and produce overly narrow confidence limits for trend estimates. Partial explanations for this underestimation include failure to account for serial autocorrelation, or a lack of independence, in time series data (Ryding et al. 2007) as well as an assumption of no process variance, or annual variation about the long-term mean growth rate (Humbert et al. 2009). We used code developed by Humbert et al. (2009) to apply exponential growth state-space models for estimating growth rates and confidence intervals while accounting for both process noise and observation error contained within these counts. While confidence intervals using these methods are likely to be wider, they have been shown to be far superior to conventional methods in best representing the overall uncertainty in count data sets.



**Figure 2.** A radio-collared moose (F332) and the shadow of a helicopter above in the Big Hole Valley, Montana, March, 2014.

Point estimates of mean growth rates ( $\hat{r}$ ) were negative for 12 of the 16 hunting districts. However, confidence intervals surrounding mean growth rates overlapped 0 in all districts except HD 332, for which the estimated trend was significantly positive (Figure 3).



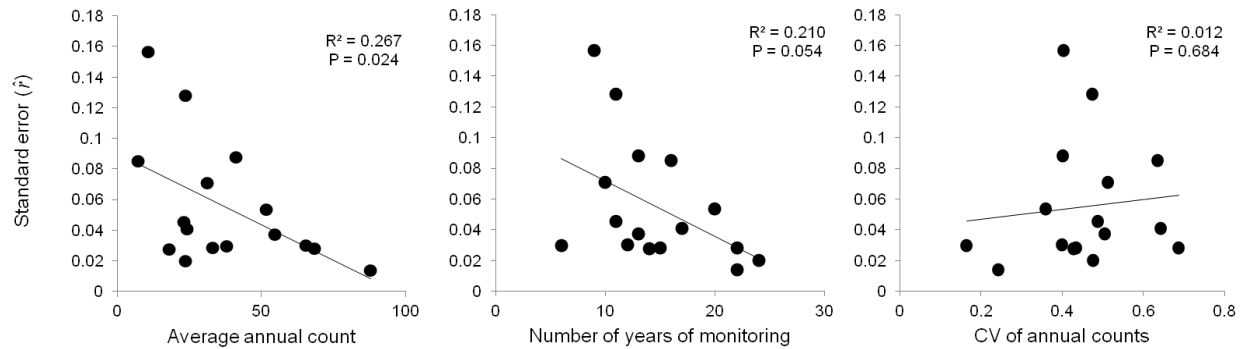
**Figure 3.** *Intrinsic population growth rate estimates ( $\hat{r}$ ), with 95% confidence intervals, and sample sizes of aerial counts (labels) for moose populations in 16 hunting districts of Montana during portions or all of 1985–2013.*

### 1.1.3. Drivers of uncertainty about trends from aerial counts

To better understand the variability in precision of estimates across hunting districts, we treated the standard errors of growth rate estimates from 1.1.2 as a response variable for modeling which parameters were most responsible for improved precision. We used multiple linear regression to assess the effect of the number of annual surveys, the total duration spanned by surveys, the average count recorded during surveys, and the coefficient of variation (CV) of surveys upon standard error of growth rate estimates. Because there were few cases of replicated surveys within a given year, we estimated mean and standard deviation values of repeated survey counts within three-year moving windows. We then estimated the CV of each set of 3 consecutive counts, and averaged these to estimate the mean CV of surveys within each hunting district. Average CV estimates per HD were likely to be conservative overestimates of sampling variance, because they assume all variation in counts within a three-year span was entirely the result of variation in visibility rather than true changes in abundance (i.e., process variance).

The best model ( $\Delta\text{AIC} = 0$ ,  $R^2 = 0.267$ ) explaining the precision of growth rate estimates included only a negative effect ( $\beta = -0.00093$ ,  $P = 0.024$ ) of the average number of moose counted during annual surveys per HD upon the variance of trend estimates (Figure 4a). There was also support ( $\Delta\text{AIC} = 0.13$ ,  $R^2 = 0.348$ ) for a model including both a negative effect of average counts ( $\beta = -0.00072$ ,  $P = 0.121$ ) as well as a negative effect ( $\beta = -0.0024$ ,  $P = 0.227$ ; Figure 4b) of the total number of survey-years per HD. Surprisingly, the estimated CV of annual counts was not predictive ( $\beta = 0.032$ ,  $P = 0.684$ ; Figure 4c) of the overall precision of growth rate estimates, though estimated CVs of surveys were somewhat high (mean = 0.45, range=0.16–0.69) relative to typical CVs targeted by surveys (e.g., target of <0.20 by Alberta SRD).





**Figure 4.** Relationships between the precision (standard error) of growth rate estimates per HD and other parameters describing the aerial count data in those HDs, such as the average number of moose counted, the number of surveys conducted, and the estimated CV of consecutive surveys, Montana, 1985–2013.

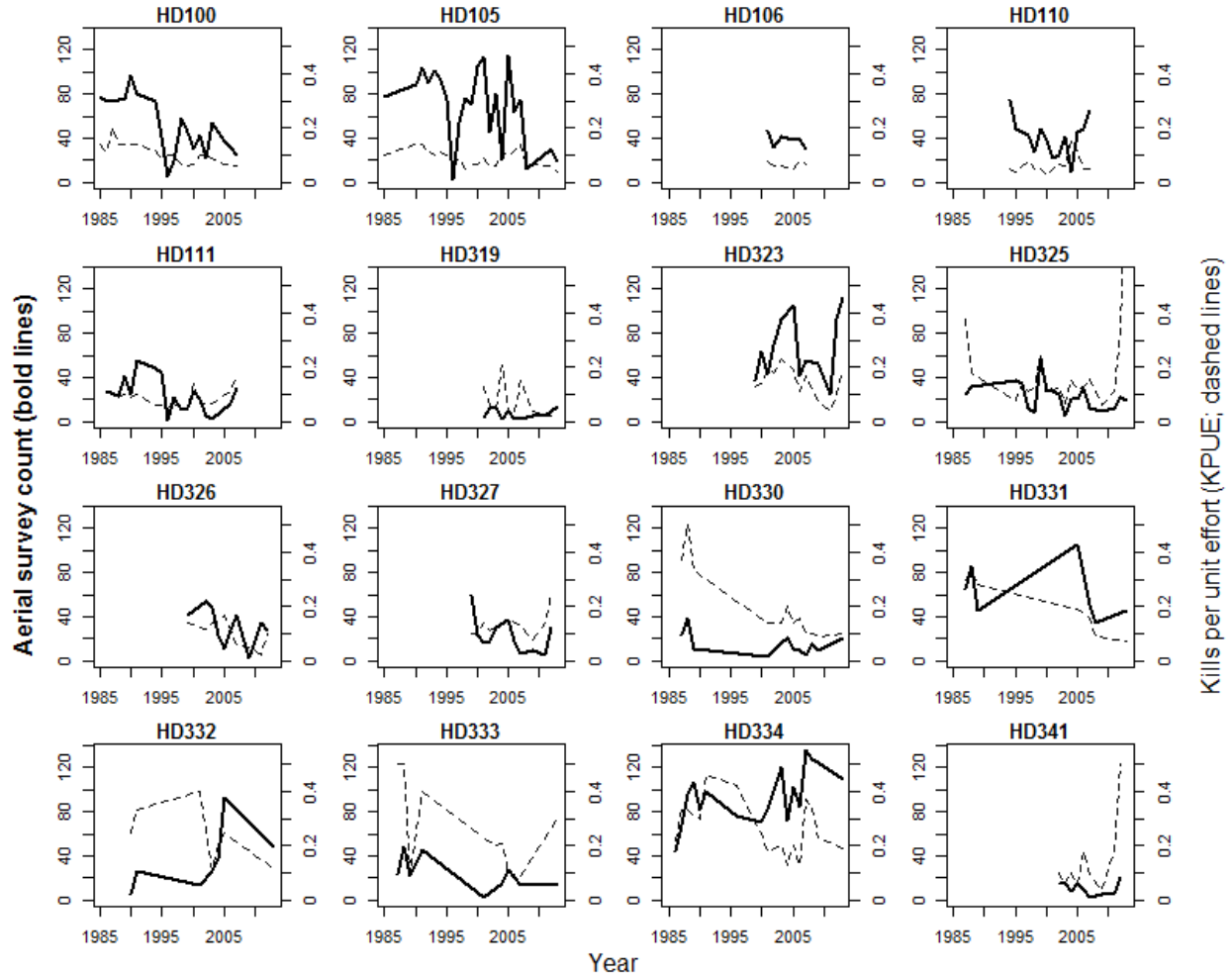
#### 1.1.4. Evaluating KPUE as an index of count-based trends

Lastly, we paired aerial survey data with harvest data collected during 1985–2013 across 16 HDs to assess the relationship between KPUE during a given hunting season and aerial survey counts conducted within the subsequent winters of the same biological year (Figure 5). The relationship between KPUE and abundance has been characterized with the function:

$$KPUE = \alpha N_t^\beta \quad [\text{equation 1}],$$

where  $\alpha$  is the standardized coefficient of catchability and  $\beta$  is the catchability exponent (Cooke and Beddington 1984, Hatter 2001).

It is often inherently assumed that  $\beta=1$ , which parameterizes a positive and linear correlation between KPUE and abundance, with changes in KPUE assumed to be directly proportionate to those in abundance. However, situations where  $\beta<1$  (“hyperstability”) can occur when harvest is particularly efficient or concentrated in areas of high abundance, such that KPUE estimates would optimistically convey stability in a population that is in fact declining. Conversely,  $\beta>1$  (“hyperdepletion”) might convey pessimistic trends if some large proportion of the population is functionally unavailable for harvest, as might occur with harboring of animals on un hunted private land. This relationship between KPUE and abundance has rarely been tested with ungulates. Hatter (2001) found evidence of hyperstability when using KPUE to monitor declining populations of both black-tailed deer (*Odocoileus hemionus columbianus*;  $\beta=0.58$ ) and moose ( $\beta=0.46$ ). Similarly, Ueno et al. (2014) found a less than proportional relationship (i.e.,  $\beta<1$ ) between an index of moose seen per unit effort (SPUE) and moose density. Contrary to this, Bowyer et al. (1999) found that KPUE actually increased with declines in abundance in a low density population ( $\beta<0$ ). Clearly, the value of  $\beta$  describing this relationship between harvest indices and abundance for any particular wildlife population would have strong implications for managers wishing to interpret population trends with KPUE (or SPUE) alone.



**Figure 5.** Time series of moose aerial counts (bold) and moose hunter kills per unit effort (dashed; KPUE) in 16 hunting districts of Montana during portions or all of 1985–2013.

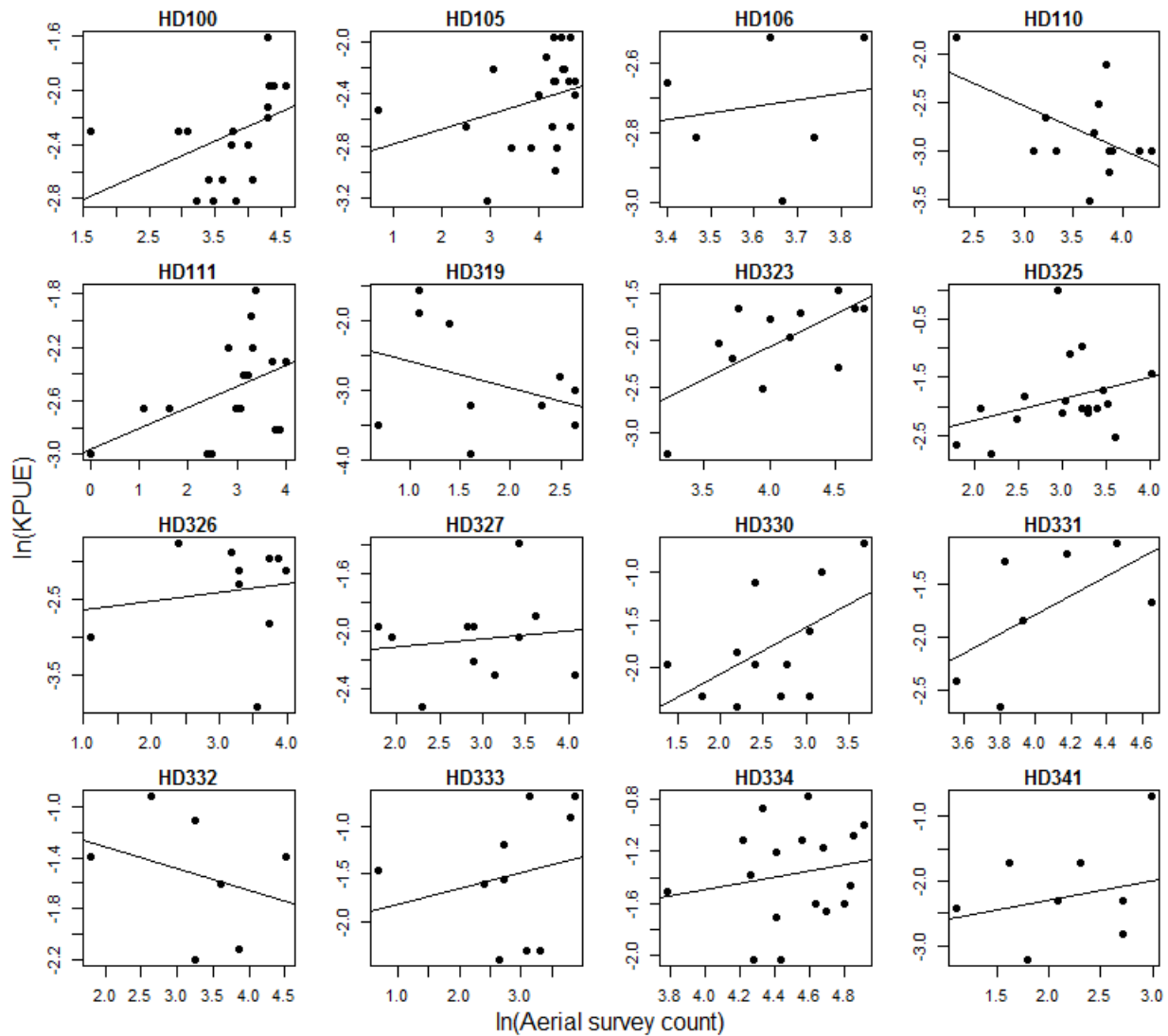
We modified the approach of Hatter (2001), and used general liner mixed-effects models (GLMMs) in the form of:

$$\log_e(\text{KPUE}_{ij}) = \log_e(\alpha) + \beta(\log_e[N_{ij}]) + \log_e(\gamma_{aj}) + \gamma_{\beta j}(\log_e[N_{ij}]) \quad [\text{equation 2}],$$

to generate both fixed-effect, pooled estimates of the catchability coefficient,  $\alpha$ , and catchability exponent,  $\beta$ , as well as random intercepts  $\gamma_{aj}$ , adjusting  $\alpha$  per  $\text{HD}_j$ , and random coefficients  $\gamma_{\beta j}$ , adjusting  $\beta$  per  $\text{HD}_j$ . We included random effects in the models to account for repeated (and thus correlated) observations within HD, as well as the potential for different relationships between counts and KPUE across HDs. We  $\log_e$  transformed KPUE and aerial count data ( $N$ ) prior to modeling, and then conducted linear regression on transformed values. We then back-transformed model fixed- and random-intercepts to calculate  $\alpha$ , and estimated back-transformed predictions of KPUE using equation 1. We used AIC to evaluate support for a model that included both random intercepts (catchability coefficients,  $\gamma_{aj}$ ) and random coefficients (catchability exponents,  $\gamma_{\beta j}$ ) for each hunting district compared to a model that included only random intercepts and a single, population-level fixed effect estimate of  $\beta$ . We estimated marginal and conditional  $R^2$  values following Nakagawa and Schielzeth (2013) to characterize

the percentage of variance explained both by fixed effects only and by the combination of fixed and random effects, respectively.

In total we modeled the relationship between KPUE and aerial survey counts using 202 district-years of data for which both aerial survey and harvest statistic information were available (Figure 6). Visual plots revealed a positive relationship between the natural log of aerial counts and that of kills per unit effort for most but not all HDs (Figure 7), suggesting that decreases in KPUE were generally corroborated by decreases in counts.



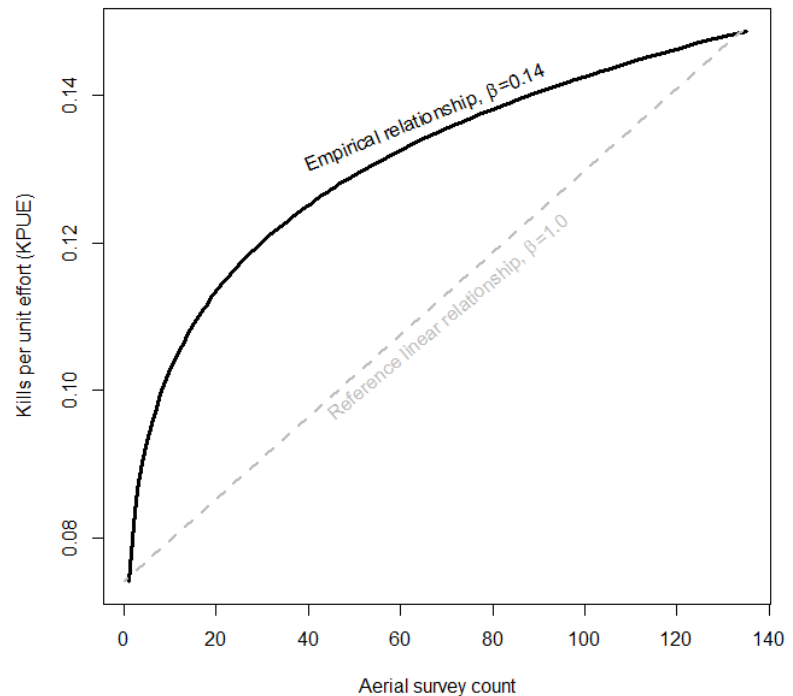
**Figure 6.** Linear regression fits assessing the relationship between the natural logs of hunter kills per unit effort (KPUE) and aerial survey counts 16 hunting districts of Montana during portions or all of 1985–2013.

Despite visual evidence of variation across HDs, the best model included random intercepts for each HD but a single fixed-effect estimate of the catchability exponent ( $\beta = 0.142$ ,  $SE = 0.048$ ), resulting in the prediction:

$$KPUE = (0.074 + \gamma_{aj}) * N_t^{0.142} \text{ [equation 3],}$$

where  $\gamma_{aj}$  is the random intercept unique to each HD.

**Figure 7.** *The best model explaining the relationship between KPUE and aerial counts included a single, overall, estimate of the catchability exponent  $\beta=0.14$  across 16 hunting districts of Montana during portions or all of 1985–2013. This estimate parameterizes a situation of hyperstability, whereby population trends estimated using KPUE alone could show evidence of population stability while underlying aerial count data are in fact in decline.*



While significantly positive, this point estimate of the catchability coefficient (0.14) was also significantly  $<1$ . This suggests some level of hyperstability (Figure 7) when monitoring Montana moose with KPUE, such that early population declines would be less likely to be detected by trend estimates derived from KPUE as compared to those derived from aerial count data. The conditional (i.e., combined)  $R^2$  ( $R^2_{GLMM(c)} = 0.454$ ) indicated that the best model explained 45% of the variation in aerial counts. However, the much lower marginal  $R^2$  ( $R^2_{GLMM(m)} = 0.044$ ) indicated that the great majority of this explanatory power came from mean differences among HDs explained by random intercepts, and that the fixed effect of KPUE itself explained only 4.4% of the total variation.

#### 1.1.5. Discussion –

Our results reveal a great deal of statistical uncertainty surrounding population trends of moose in Montana when using aerial-based minimum count data, despite time series including an average of 14.7 annual counts, per district. While point estimates of  $r$  suggest widespread declines in moose numbers, these data are generally insufficient to conclude that populations have in fact declined with great confidence. Much of this uncertainty likely stems from the inherent challenges of lacking statistical power when surveying small populations with low sightability (MacKenzie et al. 2005). Supporting this statement, we found the magnitude of aerial counts to be the biggest driver of precision in growth estimates among HDs. Not surprisingly, biologists should expect better estimates of trend in larger or more visible



populations and poor estimates of trend in small populations or those with poor sightability. Given that variation in sightability is the underlying driver of variation about counts, it should also be noted that long-term changes in sightability, such as might occur with the regeneration and aging of forests in which moose reside, could also produce apparently declining population trends without any underlying changes to abundance.

In our case, relative differences among HDs in terms of coefficients of variation among consecutive annual counts were not predictive of the precision of growth rate estimates. This somewhat contradicts a previous recommendation to prioritize minimizing the CVs of individual abundance estimates over increasing the frequency of the counts (Humbert et al. 2009), yet the discrepancy may have arisen for a few reasons. Firstly, our CV estimates are based on an unlikely assumption that consecutive surveys conducted over a three-year period were repeated samples of a population of the same size. Second, the majority of our minimum count data may be too imprecise to show an effect of estimated CV on precision of growth rate estimates. Harris (1986) recommended that multiple surveys per year be conducted in situations where the variability of any single count cannot be reduced enough to produce sufficiently precise trends.

Our results also suggested that changes in KPUE were not directly proportionate to those of aerial counts. This provides some evidence supporting hyperstability, wherein populations that are in fact declining may appear stable when estimating trends with this metric alone. While few studies have assessed this relationship through direct estimation of the catchability exponent  $\beta$ , there is evidence to support both proportionate (Solberg and Sæther 1999, Kindberg et al. 2009, Cumberland 2012) and disproportionate relationships between trends measured via indices such as KPUE and trends in abundance (Bowyer et al. 1999, Hatter 2001, Ueno et al. 2014). The rigor of our test is hampered somewhat by the relative lack of precision of count estimates. A lack of proportionality between minimum aerial counts and true abundance could also be driving our results, as well as the relatively low proportion of variance (4.4%) in counts explained by KPUE. KPUE may also be subject to variation in human behavior, given that hunter effort in a trophy hunting scenario may not accurately reflect that needed simply to be successful. Lastly, changes in the number of permits available may also affect this relationship. For example, a typical management strategy might be to reduce permits for a declining population, but changes in permits might also affect success rates. Future analyses will also include consideration of permit numbers, as well as testing another response variable, moose seen per unit of hunter effort (SPUE), which may provide a similar index without being confounded by trophy hunting behavior.

## 1.2. Applying occupancy models to hunter sightings data

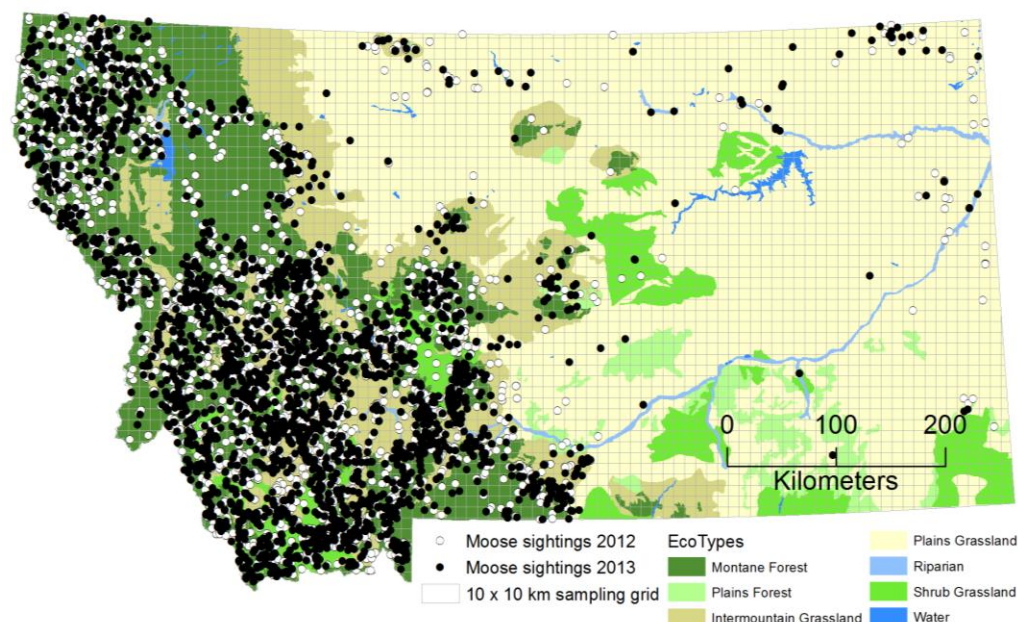
### 1.2.1. Background

Many wildlife monitoring programs have recently been developed under the framework of occupancy modeling, which allows biologists to control for the probability of detecting animals while seeking to estimate their distributions. Occupancy modeling essentially applies a mark-recapture framework to documenting the presence of unmarked individuals of a given species within spatial sampling units and over multiple sampling sessions (MacKenzie et al. 2002). Multiple years of occupancy data can also be applied towards estimating trends in spatial distribution (MacKenzie et al. 2003).

Because it does not require marked animals, occupancy modeling lends itself well to data collected by various means, including citizen science data collected by the general public (Hochachka et al. 2012, van Strien et al. 2013). For example, Rich et al. (2013) recently estimated occupancy models for wolves in Montana by collecting hunter sightings of wolves and subdividing them into sampling sessions according to each week of the five-week hunting season. During 2012–2013 we have similarly collected hunter sightings data for moose, with the intention of evaluating the potential for using occupancy modeling to monitor statewide trends in moose presence and distribution.

### 1.2.2. Methods

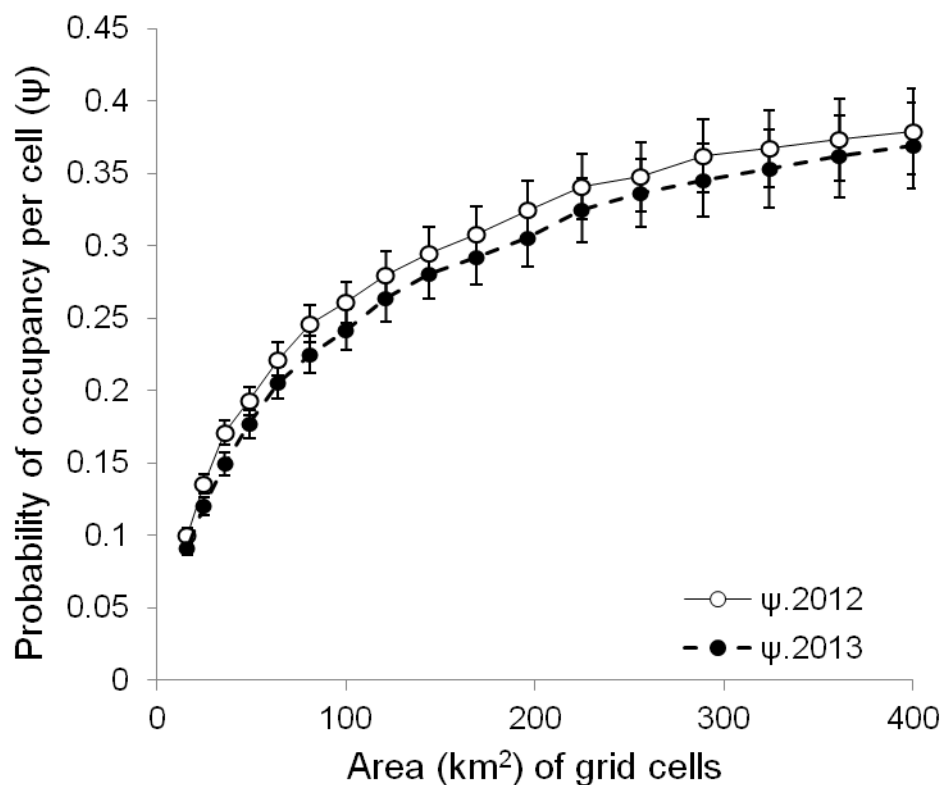
Each year MFWP staff conduct phone surveys of a large sample of resident deer and elk hunters in Montana to facilitate estimation of various hunter harvest and effort statistics. Following the 2012 and 2013 hunting seasons, a subsample of these hunters were also asked to describe the location and group size of any moose sightings that occurred while hunting. These efforts resulted in 5,782 and 4,046 statewide moose sighting locations in 2012 and 2013, respectively (Figure 8).



**Figure 8.** *Moose sightings collected using phone surveys of deer and elk hunters and an example 10 x 10 km grid for sampling statewide occupancy during the fall, 2012 and 2013, Montana.*

We are still in the preliminary stages of applying occupancy models to these data, but have done some initial analyses to assess variation in occupancy and the probability of detection. An important first step in conducting occupancy models is the selection of a sampling grid, as the dimensions of the pixels within this grid have the potential to affect results (Efford and Dawson 2012). To objectively assess the optimal pixel size given our data, we conducted occupancy analyses, estimating the occupancy rate and probability of detection, using a variety of pixel sizes from 4 x 4 km – 20 x 20 km (Figure 9). While a specific inflection point isn't entirely clear, there is evidence of somewhat of an asymptote around 100 km<sup>2</sup>, above which the estimated rate of occupancy is less affected by changes in cell size. More importantly, the estimated trend in occupancy ( $\psi_{t+1} / \psi_t$ ) appears somewhat stable with changes in cell size, but we will assess this in more depth in future studies. We will also use data from radio-collared animals to assess the range of variation in seasonal home range size during the sampling period.

Once a cell size is selected, follow-up analyses will include improvement of occupancy predictions by adding covariates likely to drive spatial variation in both occupancy (e.g., habitat type) and probability of detection (e.g., hunter effort or road access) within each grid cell.

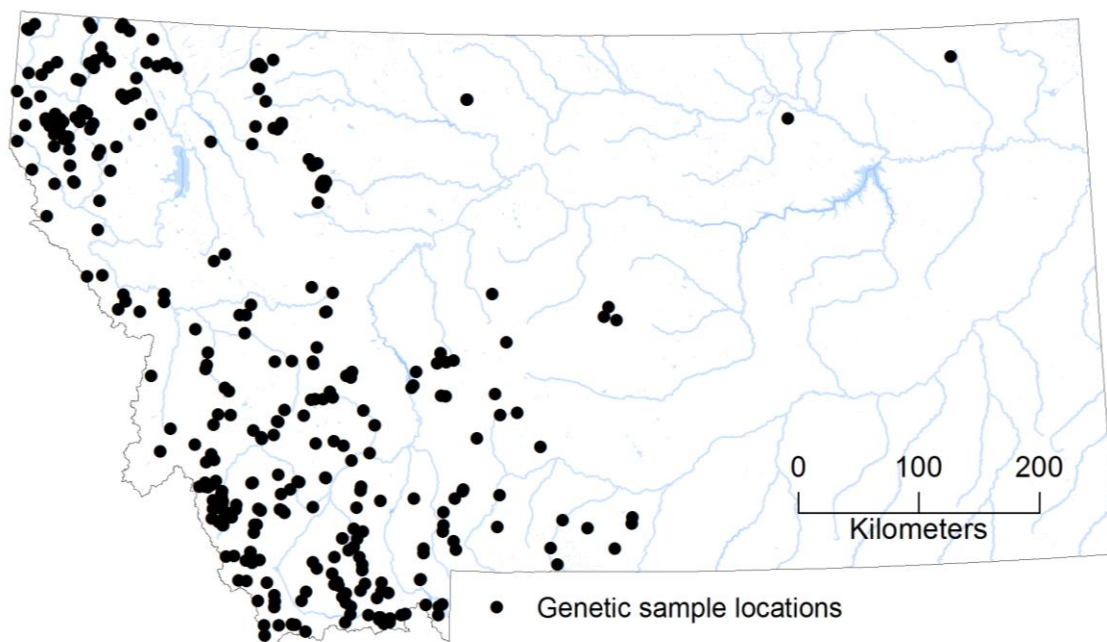


**Figure 9.** Changes in the estimated rate of statewide moose occupancy ( $\psi$ ) with changes in the dimensions of the grid cells overlaid across the state.

### 1.3. Sampling statewide moose genetic population structure.

An assessment of moose population genetic structure in Montana is lacking. However, such an assessment could provide information on population connectivity and have implications for designating biologically relevant population units for future management and monitoring. Assessment of genetic variation in Montana's moose may also have implications for taxonomy of subspecies. Moose within Montana and the rest of the US Rocky Mountains have historically been classified as Shiras moose (*A. a. shirasi*), a subspecies whose range is believed to extend northward into a zone of intergradation with the northwestern subspecies (*A. a. andersoni*) in Alberta and British Columbia (Peterson 1952). While mitochondrial haplotypes have generally upheld some level of differentiation between Shiras moose in Colorado and representative samples from other subspecies (Hundertmark et al. 2003), such methods have not been applied to evaluate moose in Montana. Particular interest in subspecies distinctions has arisen recently with anecdotal evidence of immigration of moose into north-central and -eastern Montana from expanding populations in southern Alberta and Saskatchewan.

The initial sampling of moose tissue through statewide hunter harvest is an important first step towards addressing this information gap. During the 2012 and 2013 hunting seasons, we have sent mailings to all moose permit-holders in an effort to recruit hunters as collectors of data and samples across the statewide moose population. This effort, combined with opportunistic sampling by area biologists and sampling from live-captures has resulted in 381 genetic samples statewide during 2012-14 (Figure 10). We will collect additional samples during the 2014-15 season and initial laboratory analyses on a subset of samples has already begun.

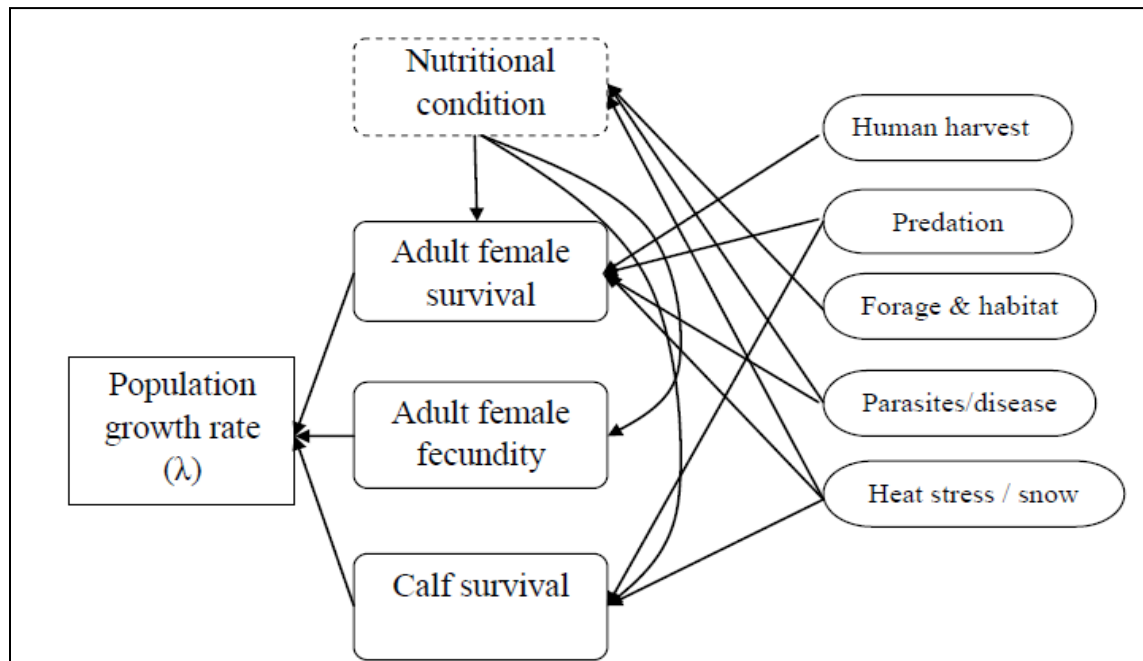


**Figure 10.** Locations of genetic samples (tissue and blood) from moose collected during 2012–2014.

## Objective #2: Monitor moose vital rates and potential limiting factors

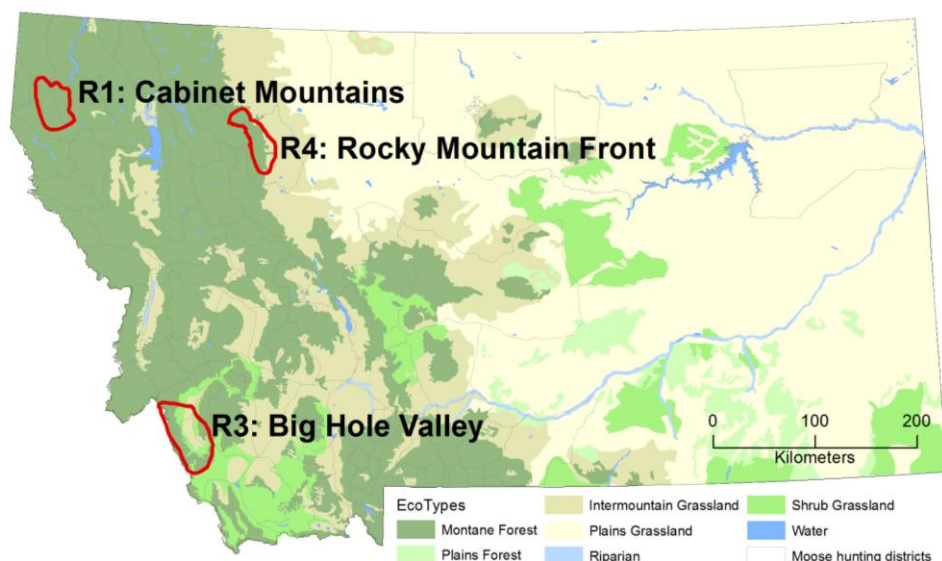
### 2.1. Background

This research project is designed to provide inferences regarding moose population dynamics using a comparative study design. This involves replicating field methods at multiple study areas that contrast in the hypothesized ecological drivers of interest (Figures 11, 12). Monitoring moose vital rates, concurrently with potential limiting factors, will allow assessment of the importance of specific vital rates to population growth and the factors influencing those vital rates.



**Figure 11.** Ecological drivers hypothesized to influence specific moose population vital rates and ultimately population growth.

**Figure 12.** Three study areas and the associated MFWP region names (1, 3, & 4) where we are using radio-telemetered animals to assess moose vital rates and limiting factors in Montana.





## 2.2. Animal capture and handling

In March of 2014 we worked with a contracted helicopter capture company (Quicksilver Air), local landowners, and pilots to conduct captures and increase the sample of monitored moose. A total of 37 adult females were captured in the 3 study areas this March. Moose were fit with either VHF (LMRT-4 model) or GPS (Lifecycle model) radio-collars from Lotek Wireless, Newmarket, Ontario. As of August 2014 a total of 62 radio-marked adult female moose are being monitored, after accounting for 2 years of capture efforts and mortalities (Table 1). This moves the study closer to the ultimate goal of maintaining a sample of 30 marked adult females in each study area. A target sample size of 30 individuals/study area was previously determined to cost-effectively achieve precision in age-class specific annual survival estimates, while minimizing capture and monitoring costs.

**Table 1.** Sample sizes of adult female moose marked with VHF or GPS radio-collars in 2014 by study area. Also shown is the total number of adult females being monitored as of August 2014.

	Study Area		
	Cabinet-Fisher	Big Hole Valley	Rocky Mtn Front
Moose captured in 2014 (Collar types deployed)	7 (5 VHF, 2 GPS)	21 (16 VHF, 5 GPS)	9 (3 VHF, 5 GPS)
Moose alive and being monitored, as of Aug 2014	16	27	19

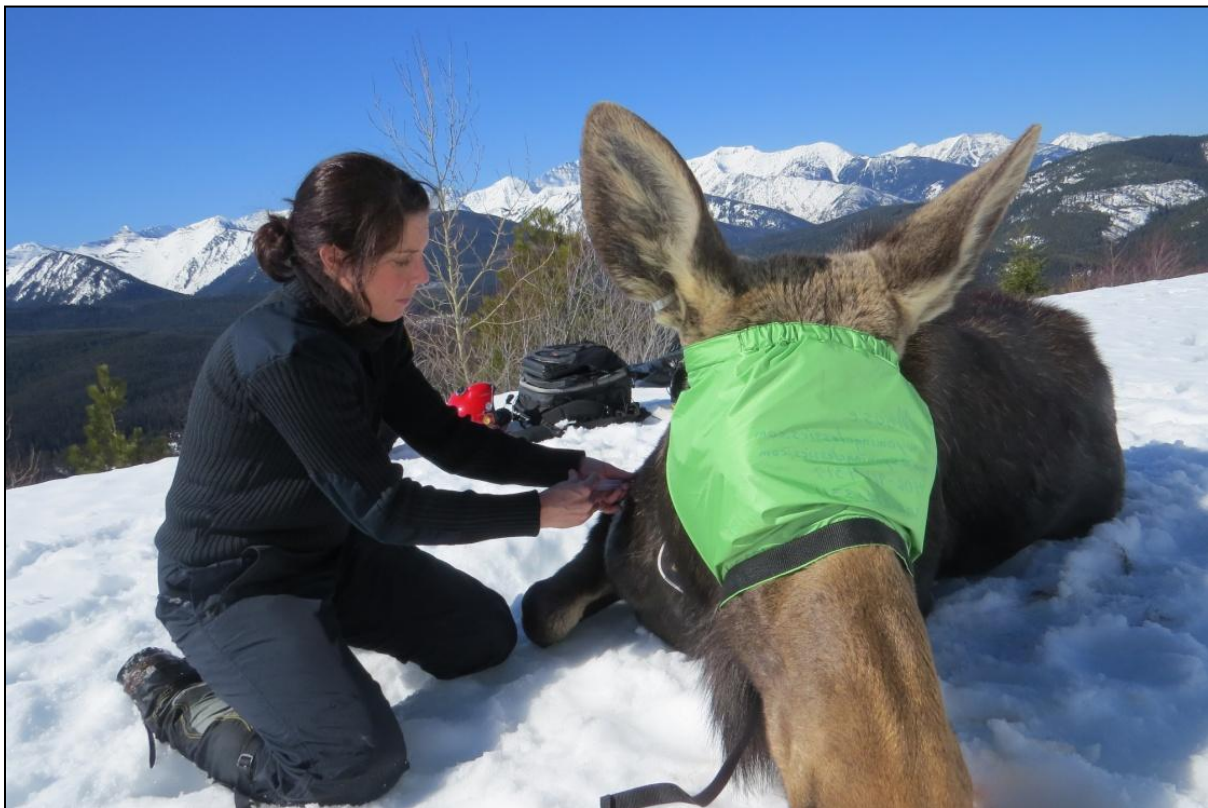
All individuals were captured via helicopter darting using carfentanil (3.3–3.9 mg/adult) and xylazine (50 mg/adult). Carfentanil was reversed with 350–400 mg of naltrexone administered intramuscularly and xylazine reversed with 400–600 mg tolazoline. Animals were kept in sternal recumbency with head higher than the body to avoid rumen regurgitation and aspiration (Kreeger 2000), and baseline temperature, pulse, and respiratory rate (TPR) values were recorded following Franzmann et al. (1984). All capture and handling procedures followed protocols approved by Montana Fish, Wildlife & Parks Animal Care and Use Committee (Permit # FWP12-2012). Despite efforts to minimize impacts on animals, captures have resulted in 4 mortalities out of 73 captures (5.5%) attributed to capture-related causes. These animals were excluded from our studies of survival, and capture procedures continue to be under adaptive review.



*Figure 13. Jesse Newby checks the collar fit before releasing adult female moose F332 in the Big Hole Valley, 2014.*

Nutritional condition and total body fat of captured individuals was measured using subcutaneous rump fat thickness from ultrasonography and palpation-based body condition measurements. Live body weights are logistically difficult to record for moose given their size (181–474 kg for females  $\geq 1.5$  years old); therefore we recorded total body length, chest girth, hind foot length, and neck circumference to estimate body weight (Table 2; Franzmann et al. 1978, Wallin et al. 1996, Franzmann 2007). We also estimated relative winter tick (*Dermacentor albipictus*) loads using line transect sampling along the rump and shoulder (Sine et al. 2009).

We collected 40 ml of blood from the jugular vein in addition to fecal and hair samples, and we used a dental elevator, dental forceps, and local anesthetic (lidocaine) to extract a lower canine for aging of individuals (Swift et al. 2002, Mansfield et al. 2006). While the collection of teeth from live animals does present some concerns for short-term animal welfare (Mansfield et al. 2006), the procedure has not been shown to subsequently affect animal health and provides critical age data for mitigating potential biases in vital rate estimation with unknown ages (Prichard et al. 2012).



**Figure 14.** MFWP veterinarian J. Ramsey administering drug reversal at completion of F115 capture in the Cabinet-Fisher study area, March 2014.

**Table 2.** Mean estimated weights of captured moose in 3 study areas using estimates based both on length and girth. Franzmann et al. (1978) found a strong relationship between both length and weight ( $r = 94$ ;  $n = 502$ ) and girth and weight ( $r = 90$ ;  $n = 496$ ) for male and female moose in Alaska. However, estimated weights presented here should be interpreted with caution. Models developed from Alaskan moose may not perform well in estimating weights of Shiras moose in Montana. In addition, moose weights can vary greatly with season and this is unaccounted for within this model. Note that measurements from R4, the Rocky Mountain Front, tend to be lower than the other study areas. This is likely in part due to the larger proportion of younger animals (21 – 34 months old) captured in this area.

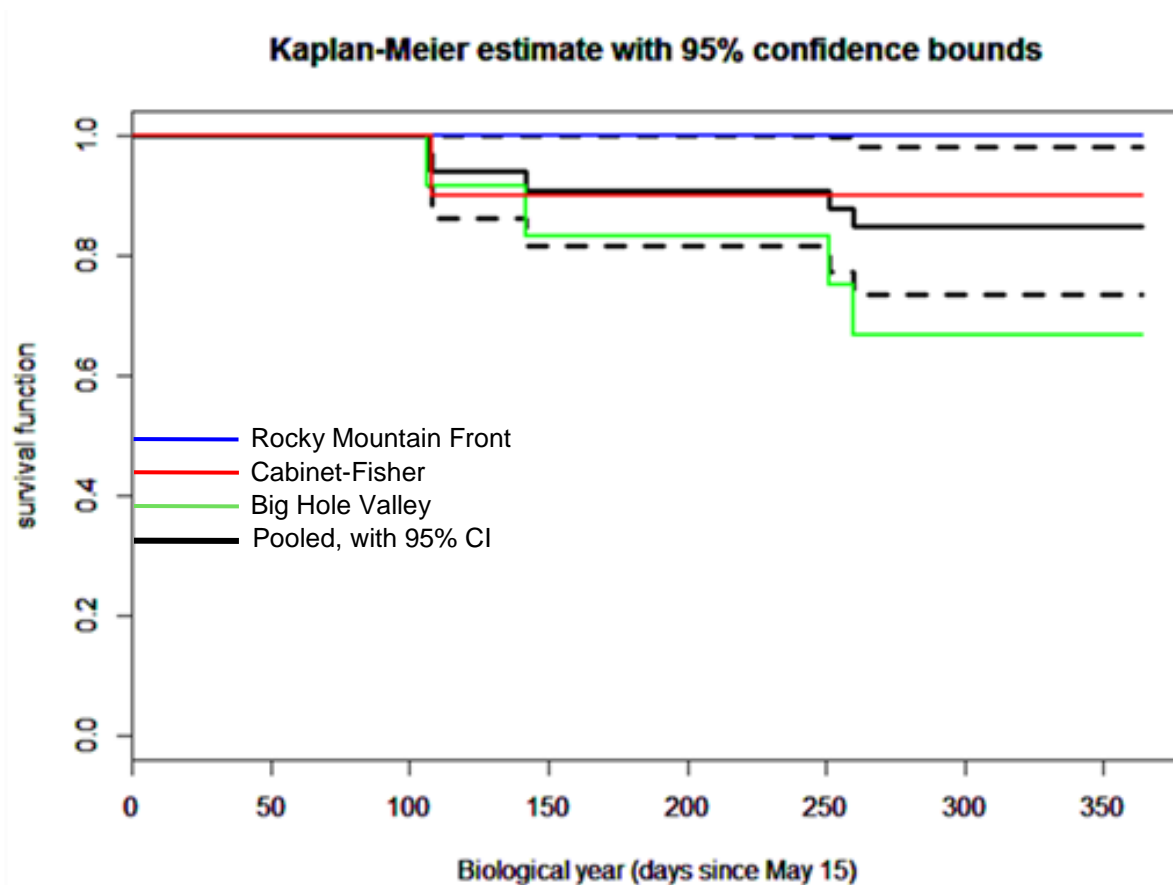
	Estimated weights (lbs.), derived from length- and girth-based equations					
	<u>Cabinet-Fisher</u>		<u>Big Hole Valley</u>		<u>Rocky Mtn Front</u>	
	Weight (length-based)	Weight (girth-based)	Weight (length-based)	Weight (girth-based)	Weight (length-based)	Weight (girth-based)
Mean	671	706	675	705	655	621
Std Dev.	54.54	63.41	51.06	75.08	79.73	91.88
N	18	18	32	31	21	21

## 2.3. Monitoring vital rates

The study of vital rates allows important mechanistic insight into the factors driving population dynamics as well as the assessment of the growth rates and population trends themselves (DeCesare et al. 2012). In May, 2014 we reached the end of our first complete biological year of monitoring since beginning the study. Below we summarize vital rate estimates (adult female survival, calf survival, pregnancy) for this biological year, and provide an update on data collected during the current biological year. Researchers in other areas have found important effects of each of these vital rates upon moose dynamics (Berger et al. 1999, Keech et al. 2000, Lenarz et al. 2010, Sivertsen et al. 2012), thus baseline estimates of each will be important for understanding dynamics in Montana.

2.3.1. Adult female survival.— Our study of adult survival to date includes 69 radio-collared adult female moose, with 2013-captured animals monitored for 1 year, 180 days, and 2014-captured animals monitored for 150 days. The average time between status (alive/dead) checks for animals wearing VHF collars was 10.4 days, and we have collected spatial locations and additional vegetation data for roughly 48% of checks.

When limiting analyses to the 2013-14 biological year, the pooled Kaplan-Meier survival estimate was 0.848 (SE=0.062), with a fairly wide 95% confidence interval of (0.735, 0.980). While sample sizes per study area during most of the biological year were quite small (range 8–12) prior to 2014 captures, there was some preliminary evidence of differences in adult survival among study areas (Figure 15). Precision of estimates will improve over the next 2 years as we continue to build in sample size.



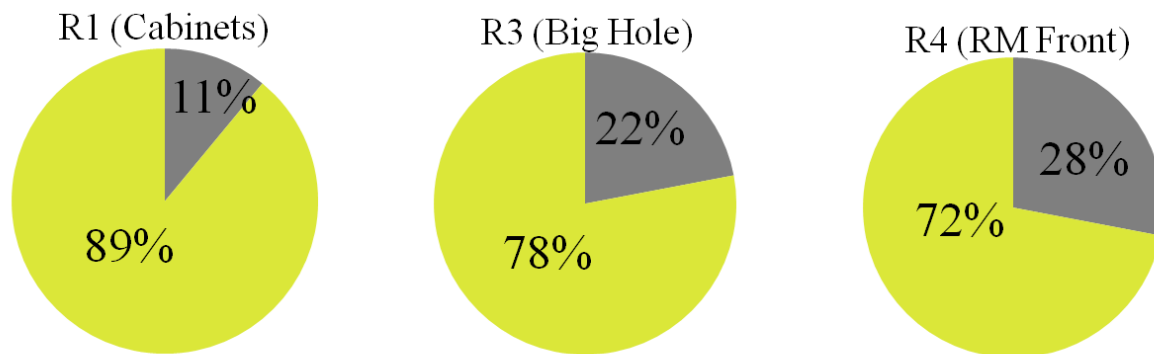
**Figure 15.** Kaplan-meier estimates of adult female survival within each study area as well as pooled across study areas, during the 2013-2014 biological year, Montana.

Our monitoring is not currently designed to directly study cause-specific mortality, but we have opportunistically collected data at mortality sites. To date, we have documented 7 mortalities of collared adult moose across all study areas: 2 in the Cabinet-Fisher, 5 in the Big Hole and 0 in the Rocky Mountain Front (Table 3).

**Table 3.** Numbers of mortalities by cause for radio-collared adult female moose documented during February 2013–July 2014, Montana.

Cause of Mortality	<u>Study area</u>		
	Cabinet-Fisher	Big Hole Valley	Rocky Mountain Front
Unknown	1	0	0
Predation, wolf	1	0	0
Disease and/or malnutrition	0	5	0

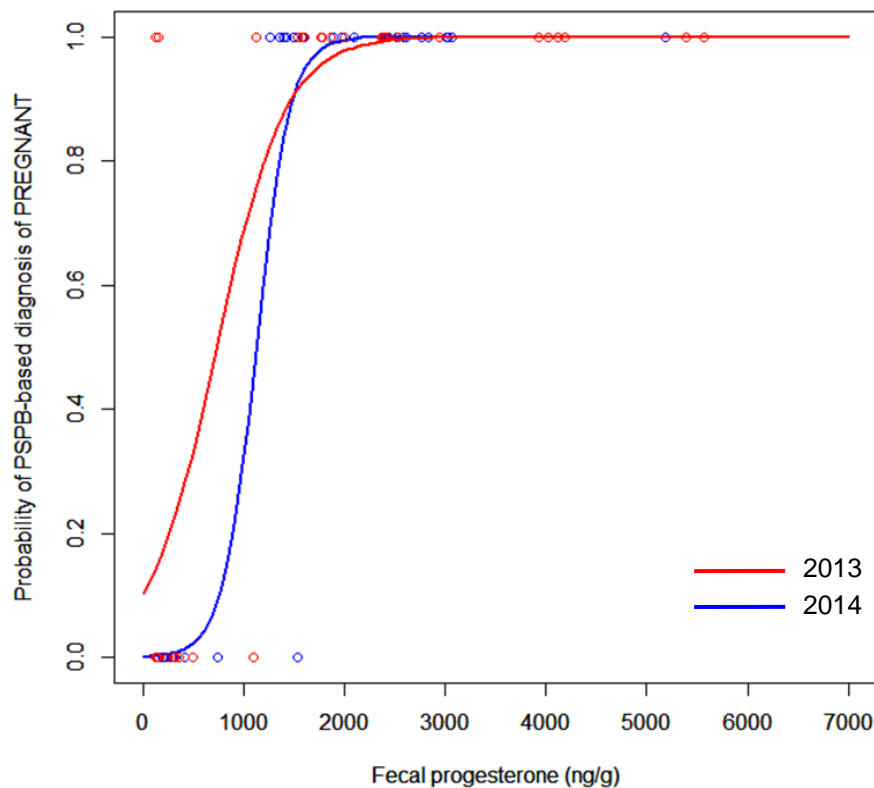
**2.3.2 Adult female fecundity.**—Fecundity for moose is the product of pregnancy rate, survival rate of fetuses to parturition, and litter size. Pregnancy of animals during winter can be estimated with laboratory analyses of both blood and scat. Blood analyses are based on the presence of a pregnancy specific protein B (PSPB) within serum samples (Huang et al. 2000). As reported by the commercial lab where analyses were conducted (BioTracking, Moscow, Idaho), this test is quite accurate in its diagnoses of non-pregnant individuals (99.9%). However, animals diagnosed as pregnant can in fact be non-pregnant 5–7% of the time. Thus some false positives may be present within the PSPB-based diagnoses. Notably, higher rates of both false positives and false negatives have been reported in wild ungulates (Testa and Adams 1998, Cain et al. 2012). Using serum-based PSPB tests alone, we documented an average adult pregnancy rate of 79.7% of adults (excludes 9 non-pregnant yearlings; Figure 16).



**Figure 16.** *Estimated adult pregnancy rates according to PSPB tests of serum, from 72 female moose captured during February–March of 2013 & 2014 in the Cabinet Mountains (N=19), Big Hole Valley (N=27), and Rocky Mountain Front (N=18) study areas, Montana. These rates exclude tests of 9 yearling moose, which were all non-pregnant, and exclude animals in subsequent years post-capture, during which they were monitored only with fecal progesterone.*

The concentration of progesterone hormone metabolites in scat samples (i.e., fecal progestagens) can also be used to detect pregnancy in moose (Berger et al. 1999, Murray et al. 2012). We measured fecal progestagen (FP) concentrations with two sampling techniques: 1) capturing animals and collecting fecal samples concurrent with blood sampling, and 2) using ground-tracking of free-ranging radio-collared moose throughout the winter (January–April) to collect fecal samples from the snow. Generally FP results were in agreement with PSPB results, and we applied logistic regression to matched data for FP and PSPB collected at the same time to model this relationship (Figure 17).





**Figure 17.** *Observed (points) and modeled (lines) relationship between fecal progesterone concentrations and pregnancy diagnoses based on PSPB in serum for moose captured in 2013 & 2014 in western Montana.*

We used logistic regression to model the probability of PSPB-based pregnancy diagnosis as:

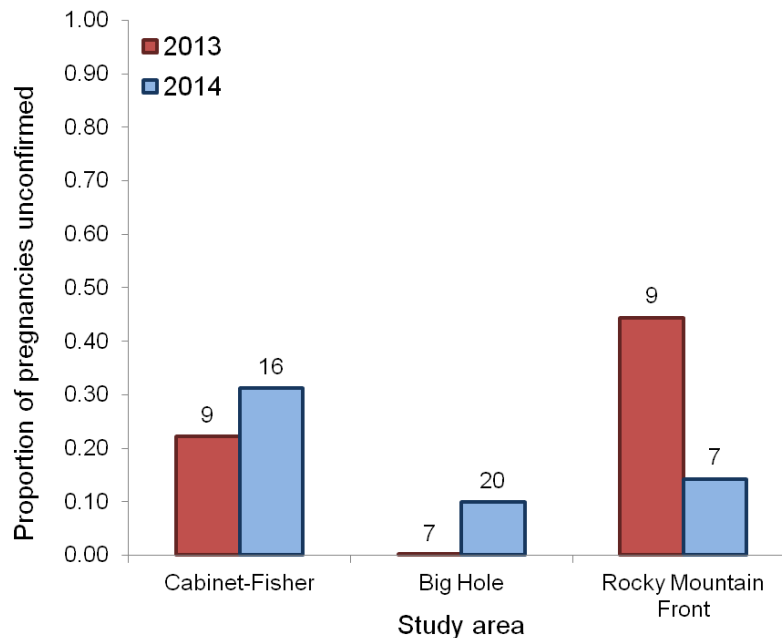
$$\Pr(PSPB_{pregnant}) = \frac{\exp(\beta_0 + (\beta_1 * FP))}{1 + \exp(\beta_0 + (\beta_1 * FP))},$$

using separate models for 2013 ( $\beta_0 = -2.17$ ,  $\beta_1 = 0.003$ ) and 2014 ( $\beta_0 = -6.77$ ,  $\beta_1 = 0.006$ ) and a single pooled model ( $\beta_0 = -3.18$ ,  $\beta_1 = 0.0036$ ). For animals not captured in 2014, we used the 2014 equation to predict pregnancy, and then compared all PSPB-based and model-based pregnancy diagnoses with field observations from weekly aerial calf-at-heel surveys in the subsequent spring.

**Table 4.** *Comparison of predicted pregnancy status for adult female moose with observed litters (i.e., calves-at-heel) for the same sample of moose suggests that up to 21% of cows predicted to be pregnant from the combination of serum and fecal assays may have incurred late-fetal or early-calf mortality before we were able to visually document presence of a calf.*

Predicted status at time of parturition	Observed litters	
	Litter	No Litter
Pregnant	54	14
Not pregnant	2	23

Using the combined PSPB and FN methods of pregnancy detection, we have diagnosed 68 moose as pregnant and 25 moose as not pregnant across 2013–2014 (Table 4). Of the 68 “pregnant” moose, we have failed to detect subsequent calves-at-heel for 14 (21%) of them, and of the 25 “not pregnant moose”, we have in fact detected calves-at-heel for 2 (8%). These might be considered maximum rates of false positives and false negatives, respectively (Figure 18). However, it is currently unclear whether false positives are the result of cows that did not give birth or cows that gave birth but whose calves perished before we could visually observe them. Preliminary evidence showing depressed FN values in early-spring suggests that some cases of fetal abortion or reabsorption likely occur, but in many cases it remains unclear.

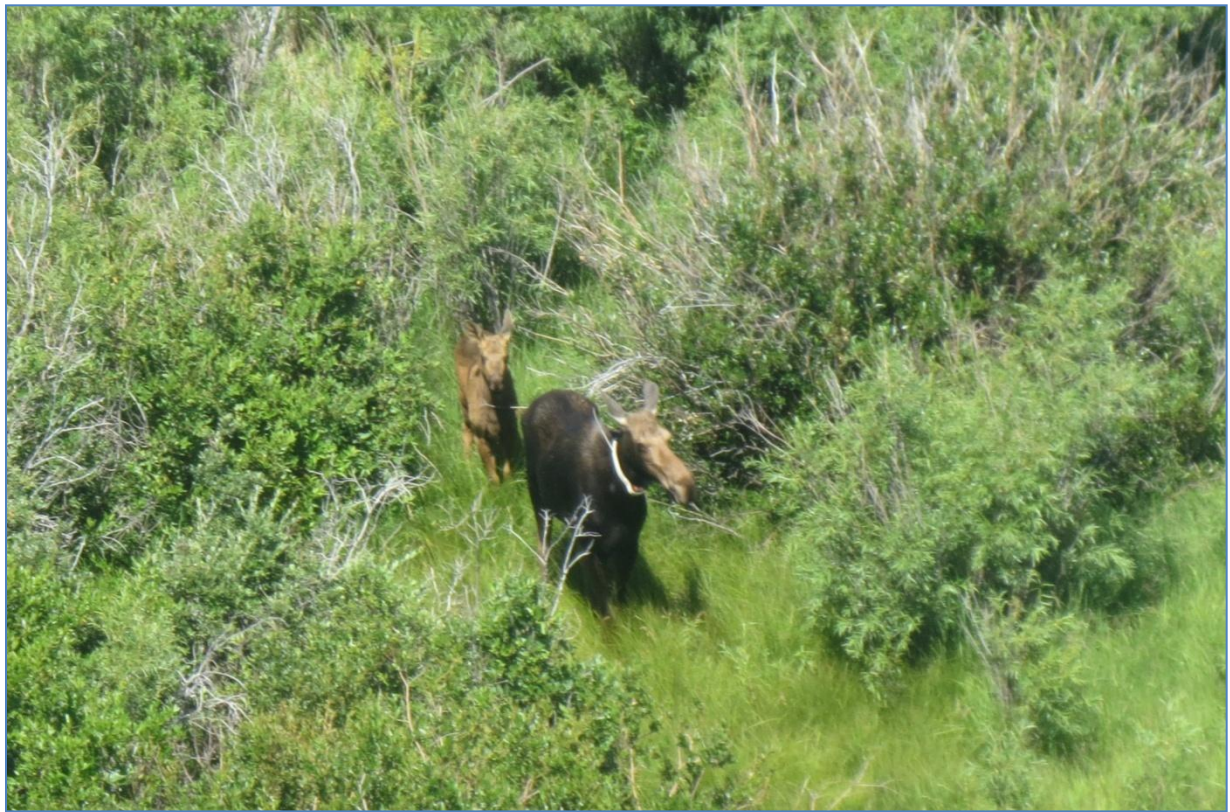


**Figure 18.** *Proportion of moose diagnosed as pregnant, but for whom a calf-at-heel was never visually documented, with total sample size of all pregnancies labelled. Unconfirmed pregnancies could be the result of false positives in the hormone assays, fetal mortality following the hormone test, or neonatal mortality following a successful birthing event but prior to our visual observation of the calf.*

Restricting results to the PSPB sampling, our overall pregnancy rate was of 79.7% of adults (excludes 9 non-pregnant yearlings), which is below the 84.2% average of adult moose pregnancy rates across North American (Boer 1992). When we compare predicted pregnancy with observed litters, it appears the realized parturition rate may be somewhat lower than this pregnancy rate. This is similar to results of other studies (e.g., Becker 2008) where parturition rates are lower than earlier winter pregnancy rates due to presumed fetal losses throughout winter. Low pregnancy rates from 48%–75% have been reported in other Shiras moose population-years (Oates et al. 2012), and combined with our results this may reflect generally lower productivity of this subspecies, or the habitat within which it resides, compared to moose further north in Alaska and Canada.

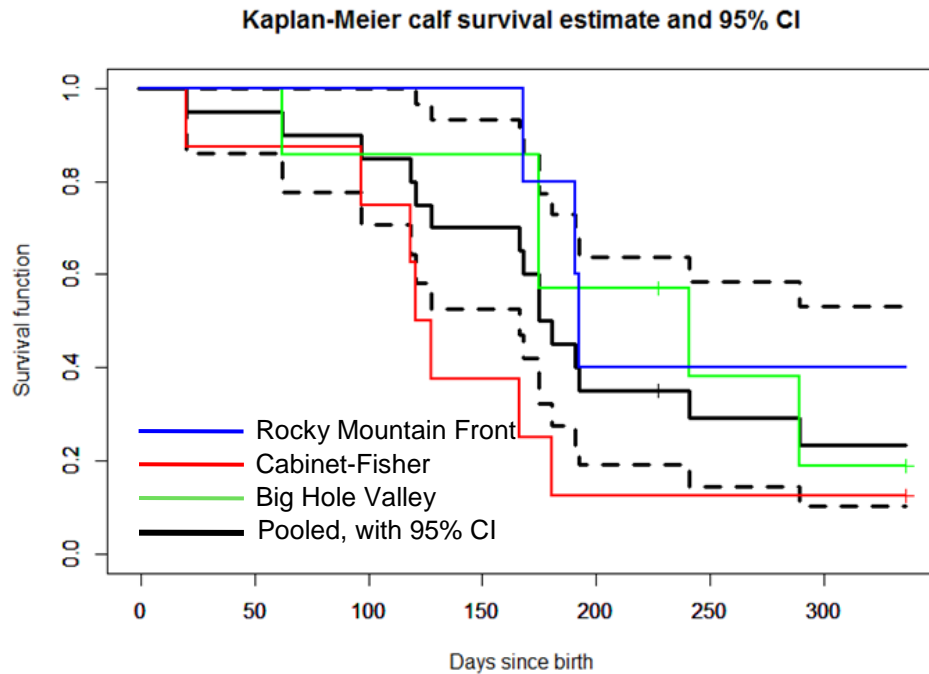
Moose are capable of giving birth to 1–3 calves, though litters are most commonly composed of either 1 or 2 calves (Van Ballenberghe and Ballard 2007). Twinning rates in North American populations can vary from 0 to 90 percent of births (Gasaway et al. 1992), with variation linked to nutritional condition (Franzmann and Schwartz 1985) and animal age (Ericsson et al. 2001). Twinning rates observed for Shiras moose appear to be relatively low (e.g., <15%; Peek 1962, Stevens 1970, Schladweiler and Stevens 1973, Becker 2008), though it is unclear if this reflects a general difference in nutrition or other locally adapted trait. Thus far our observed twinning rates are 5.3% in 2013 ( $N=19$  litters) and 2.7% in 2014 ( $N=37$  litters).

**2.3.3 Calf survival.**— We used aerial telemetry to visually search for calves-at-heel with each collared adult female at approximately weekly intervals during 15 May – 15 July. Aerial telemetry allowed efficient visual observation of cow-calf pairs with minimal disturbance, despite often dense habitats that obscure animals on the ground (Figure 19). Flights were conducted with a mix of fixed-wing and rotary-wing aircraft depending on terrain and forest cover (e.g., primarily fixed-wing in R3 and rotary-wing in R1). We documented 19 litters and 20 total calves during spring, 2013. We then monitored the fates of these calves by visually locating them with their dams throughout their first year of life. Given that some pregnant cows were never seen with calves, some proportion of calves born may have died before we were able to visually confirm them, and thus Kaplan-Meier based estimates of calf survival should be considered as somewhat optimistic estimates. In the future we will be considering other survival models such as nest success models (Dinsmore et al. 2002), which were developed to accommodate this type of unobserved mortality.



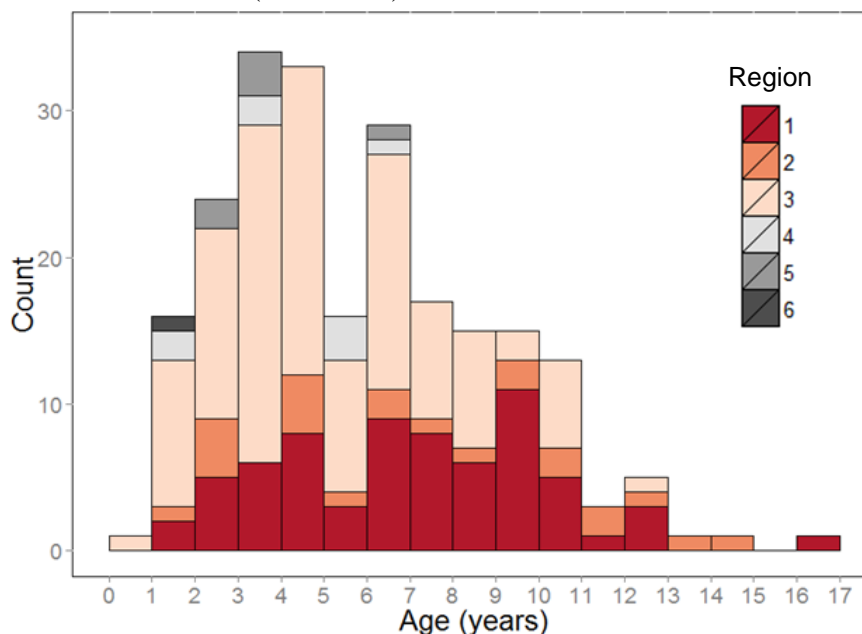
**Figure 19.** *F404 and calf on the Rocky Mountain Front, July, 2014.*

Of 20 calves monitored in 2013, 4 survived the biological year and were considered recruited into the population. This resulted in a pooled Kaplan-Meier calf survival rate of 0.233 (SE=0.098), and a 95% confidence interval of (0.102, 0.531). Individual survival curves suggest potential differences among study areas (Figure 20), yet sample sizes were quite small during this first year (N=8, 7, & 5 for Regions 1, 3, & 4 respectively).



**Figure 20.** Kaplan-meier estimates of calf survival within each study area as well as pooled across study areas, during the 2013-2014 biological year, Montana. Note this excludes calves that may have died before we were able to visually document them.

**2.3.4 Age composition.**— During the 2012 & 2013 hunting seasons we have asked moose hunters to voluntarily submit incisor teeth for cementum aging. These two years of sampling have yielded 224 aged teeth, with ages ranging from <1 to 16 years old and an overall mean of 5.3 (Figure 21). Though drawn from a sample of hunter-killed and predominately male moose, the mean age of this sample was not significantly different than that of our sample of 73 live-captured adult females (Mean=4.5).

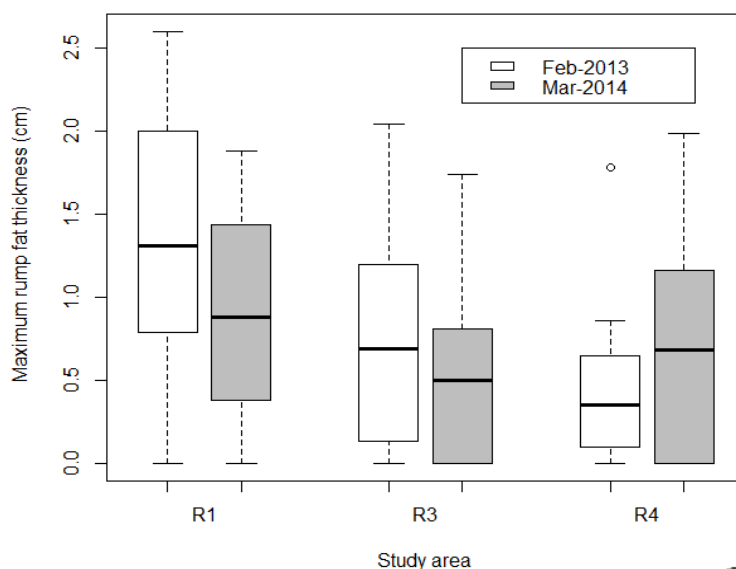


**Figure 21.** Age distribution of 224 hunter-killed moose across all regions of Montana, 2012–2013.

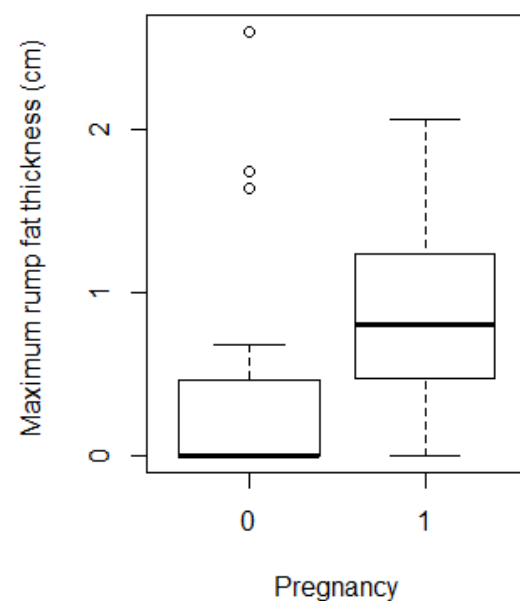
## 2.4. Nutritional condition and diet

**2.4.1. Nutritional condition.**— Nutritional condition of ungulates can impact both survival (Roffe et al. 2001, Bender et al. 2008) and fecundity (Testa and Adams 1998, Keech et al. 2000, Testa 2004), and generally provides an indication of the extent to which habitat condition may play a role in ungulate dynamics (Franzmann and Schwartz 1985, Bertram and Vivion 2002, Becker 2008). Assessment of nutritional condition across multiple study areas, potentially varying in local growth rates, will allow an initial evaluation of habitat or forage quality as a potential limiting factor.

We used a portable ultrasound (Micromaxx, Sonosite, Inc., Bothell, WA) to measure rump fat thickness in live-captured moose. Rump fat thickness has been shown to have a strong linear relationship ( $r^2=0.96$ ) with ingesta-free body fat (IFBF) in previous studies on moose (Stephenson et al. 1998). We measured rump fat thicknesses varying from 0–2.6cm, with some evidence of variation by study area (Figure 22), and by winter pregnancy status (Figure 23).



**Figure 23.** Variation in rump fat thickness of 73 female moose, ages 1–12, according to PSPB-based pregnancy status, 2013–2014.





2.4.2. Diet.— Assessing moose diet composition will provide baseline information for future work on forage availability/quality and its possible influence on nutritional condition and demography (Seaton et al. 2011). In 2013 we collected moose fecal samples in winter (February 8<sup>th</sup> – April 10<sup>th</sup>) and late summer (July 31<sup>st</sup> – September 13<sup>th</sup>) to assess diet composition in the three study areas (Table xx). Samples were initially collected directly from marked animals during capture and then collected from the relocation sites of marked individuals in later scat sampling sessions. Portions of each scat sample were combined into one composite sample per study area per season and submitted for microhistological analysis to the Wildlife Habitat and Nutrition Lab at Washington State University (Pullman, WA). Species-level analysis was conducted with 200 microscopic views per sample; providing an inventory of species foraged on by individuals approximately 1-4 days prior to collection (Wam and Hjeljord 2010). It should be noted that proportions of species found in scats cannot be interpreted as preference or even relative use because biases may exist in digestion rates and/or epidermal fragmentations. Currently no correction for these potential biases exist for moose. However, microhistological analysis of feces does have some limited ability to identify principal foods (Wam and Hjeljord 2010). This, limited information on moose diet will be useful in elucidating variation in forage use among study areas and identifying potentially important species contributing to forage resources in those areas.



**Figure 24.** *An uncollared cow moose in the Cabinet-Fisher study area, August, 2014.*

Variation in diets between studies is evident in the initial analysis of diet composition (Tables 5, 6a,b). As expected, willows (*Salix* spp.) were prominent in scat samples from all three study areas. However, the occurrence of *Salix* in microscopic views did vary considerably between areas and seasons; ranging from 18% in the Cabinet Mountains during winter to 92.6% in the Big Hole Valley during summer. Other forage species appear uniquely common in some study areas. For example, in the Big Hole Valley the use of grasses and sedges appears common in winter (combined occurrence in 200 views of winter scat sample 55%; all other seasons/areas 0.4-1.5%). The Big Hole Valley winter range is dominated by large ranches and moose commonly visit haystacks during the winter.

In contrast, the Cabinet Mountains are dominated by dense conifer forest. Here the use of conifers for forage appears more common in winter (occurrence in 200 views of winter scat sample 20.3%; all other seasons/areas 0.0-3.0%). *Vaccinium* also appears to be a commonly used forage resource in the Cabinet Mountain study area, occurring approximately in 31% of views of both winter and summer samples. Interestingly, *Vaccinium* leaves made up a large proportion of the winter scat samples in the Cabinets. Fallen leaves can be an especially important food resource for moose in some systems (Renecker and Schwartz 2007). Several of the samples collected in the Cabinets during late winter were from locations with substantial snow-free patches. It is likely that in these areas moose could exploit fallen leaves as they became increasingly available as snows melted.

**Table 5.** *Proportionate occurrence of plant groups within microscopic views of adult female moose pellets across seasons and study areas in western Montana, 2013.*

Forage group	<u>Cabinet-Fisher</u>		<u>Big Hole</u>		<u>Rocky Mountain Front</u>	
	Winter	Summer	Winter	Summer	Winter	Summer
Grasses & sedges	0.01	0	0.55	0	0	0.10
Lichen	0.01	0	0	0	0	0
Forbs	0	0	0	0	0	0.12
Shrubs	0.78	0.97	0.45	1.0	.99	0.78
Conifer	0.20	0.03	0	0	.01	0

**Tables 6A & 6B.** Results of species-level analysis of forage in moose scats collected from study areas in winter (February 8<sup>th</sup> – April 10<sup>th</sup>; Table 6A) and summer (July 31<sup>st</sup> – September 13<sup>th</sup>; Table 6B). Samples taken from female moose during capture and subsequent relocation sites. Forage items listed in order of proportion of occurrences in 200-350 microscopic views of samples. However, these proportions should not be interpreted as the proportion of forage items in moose diet in the study areas or their relative importance. Currently, a correction for differences in digestibility of forage items or variation in epidermal fragmentation between species is not available for moose scat. Furthermore, sample sizes for analysis were limited.

**Table 6A.** Winter diet

Proportionate occurrence	Cabinet Mountains N=23	Big Hole Valley N=24	Rocky Mountain Front N=20
> 20%	<i>Vaccinium</i> spp. (leaf)	<i>Salix</i> spp. (stem) <i>Carex</i>	<i>Salix</i> spp. (stem)
10 - 20%	<i>Salix</i> spp. (stem) <i>Ceanothus</i> spp. (leaf) <i>Pseudotsuga menziesii</i>	<i>Poa</i>	<i>Betula</i> spp. (stem)
5 -10%			<i>Populus</i> spp. (stem)
1 - 5%	<i>Populus</i> spp. (stem) <i>Betula</i> spp. (stem) <i>Tsuga</i> <i>Ceanothus</i> spp. (stem) <i>Thuja</i> <i>Menziesia ferruginea</i> (leaf) Unknown Grasses	<i>Populus</i> spp. stem <i>Juncus</i> <i>Symphoricarpos</i> spp. stem <i>Eleocharis</i> <i>Agrostis</i> <i>Alopecurus</i> Unknown Grasses	<i>Equisetium</i> <i>Alnus sinuata</i> (stem)
< 1%	<i>Symphoricarpos</i> spp. (stem) Lichen <i>Alnus sinuata</i> (stem) <i>Vaccinium</i> spp. (stem) <i>Carex</i> <i>Populus</i> spp. (leaf) <i>Picea</i> spp.	<i>Alnus sinuata</i> (stem) <i>Betula</i> spp. (stem) <i>Phleum</i>	<i>Shepherdia canadensis</i> (stem) <i>Abies</i> spp. <i>Cornus</i> spp. (stem) <i>Menziesia ferruginea</i> (leaf) <i>Acer glabra</i> (stem) <i>Amelanchier alnifolia</i> (stem)

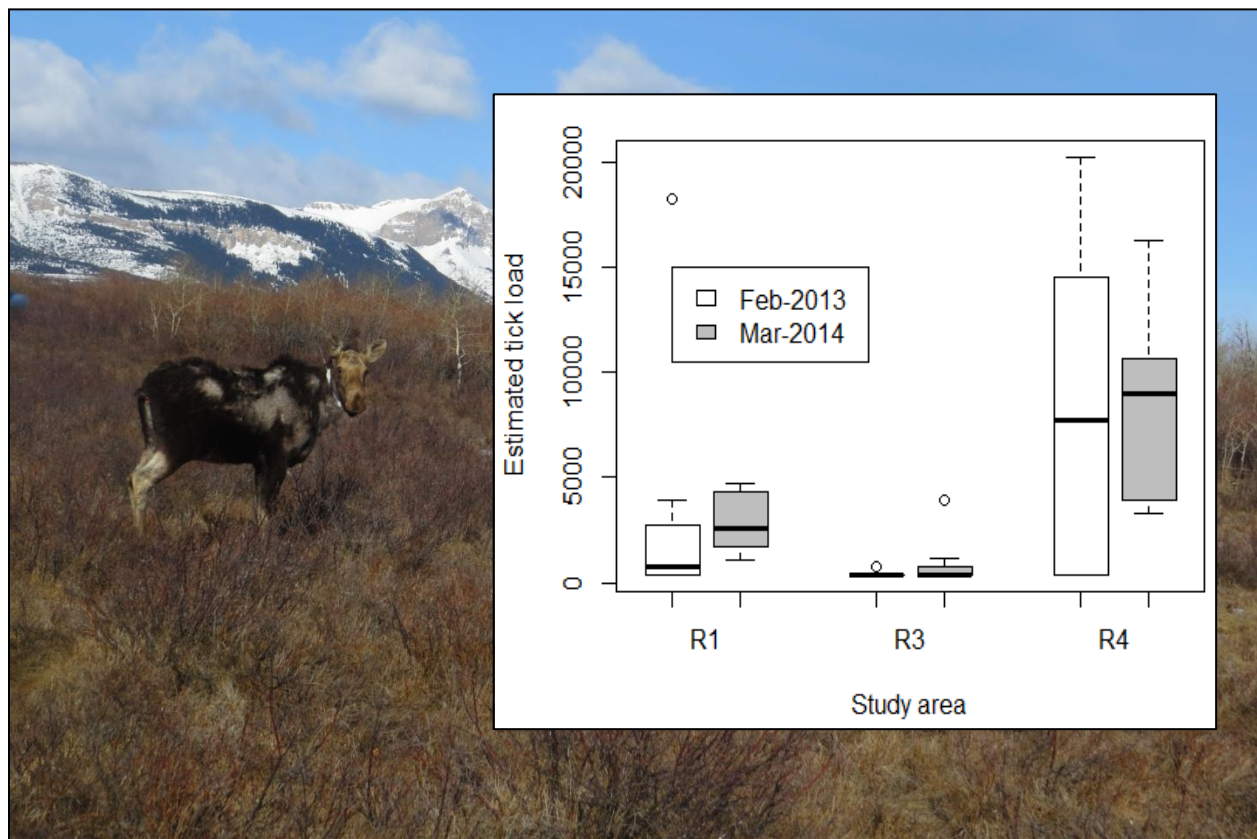
**Table 6B.** Summer diet

Proportionate occurrence	Cabinet Mountains N=6	Big Hole Valley N=12	Rocky Mountain Front N=6
> 20%	<i>Vaccinium</i> spp. leaf <i>Salix</i> spp. leaf	<i>Salix</i> spp. leaf <i>Salix</i> spp. stem	<i>Salix</i> spp. leaf
10 - 20%	<i>Salix</i> spp. stem		<i>Salix</i> spp. stem
5 -10%	<i>Cornus</i> spp. leaf <i>Vaccinium</i> spp. stem		<i>Epilobium</i>
1 - 5%	<i>Menziesia ferruginea</i> leaf <i>Symphoricarpos</i> spp. leaf <i>Ceanothus</i> spp. leaf <i>Populus</i> spp. stem <i>Alnus sinuata</i> stem <i>Betula</i> spp. leaf <i>Thuja</i> <i>Acer glabra</i> leaf Unknown Shrub leaves <i>Betula</i> spp. stem <i>Pinus</i> spp.	<i>Cornus</i> spp. leaf <i>Acer glabra</i> leaf <i>Alnus sinuata</i> leaf	<i>Carex</i> <i>Eleocharis</i> Unknown Forbs <i>Populus</i> spp. leaf Unknown Shrub leaves <i>Juncus</i> Unknown Grasses <i>Betula</i> spp. leaf <i>Populus</i> spp. stem <i>Alnus sinuata</i> leaf <i>Acer glabra</i> leaf
< 1%	<i>Sambucus</i> spp. leaf <i>Symphoricarpos</i> spp. stem <i>Alnus sinuata</i> leaf Conifer bark <i>Populus</i> spp. leaf Unknown Forbs Unknown Shrub stems	<i>Betula</i> spp. leaf <i>Populus</i> spp. leaf <i>Populus</i> spp. stem <i>Alopecurus</i> <i>Potentilla</i> spp. leaf	<i>Cornus</i> spp. leaf <i>Vaccinium</i> spp. leaf <i>Mahonia (Berberis) repens</i> <i>Linnaea</i> <i>Sambucus</i> spp. leaf Fern Unknown Shrub stems

## 2.5. Parasite and disease prevalence

Disease and parasite sampling provide valuable baseline information concerning the health and environmental stressors of moose (and other ungulates) across the state. This information is especially relevant given concerns about the effects of several parasites on moose along the southern extent of their range (Samuel 2004, Murray et al. 2006, Henningsen et al. 2012).

We estimated a range of winter tick (*Dermacentor albipictus*) densities on captured moose from 0–0.5 ticks per cm<sup>2</sup>, which translated (coarsely) to estimated total tick loads of 0–20,000 ticks per individual moose, following Sine et al. (2009). There appeared to be differences among study areas that were consistent across years (Figure 25). Tick-induced hair loss is commonly experienced by moose during March–April when ticks reach their adult life form (Mooring and Samuel 1999), though some moose in R4 showed evidence of 10–60% hair loss as early as early February.



**Figure 25.** We searched linear transects along the rump and shoulder to estimate the density of winter ticks within the coats of captured moose. In February, ticks were still predominately in their nymph life stage, but some moose did show evidence of tick-induced hair loss of 10–60%, 2013–2014, Montana.



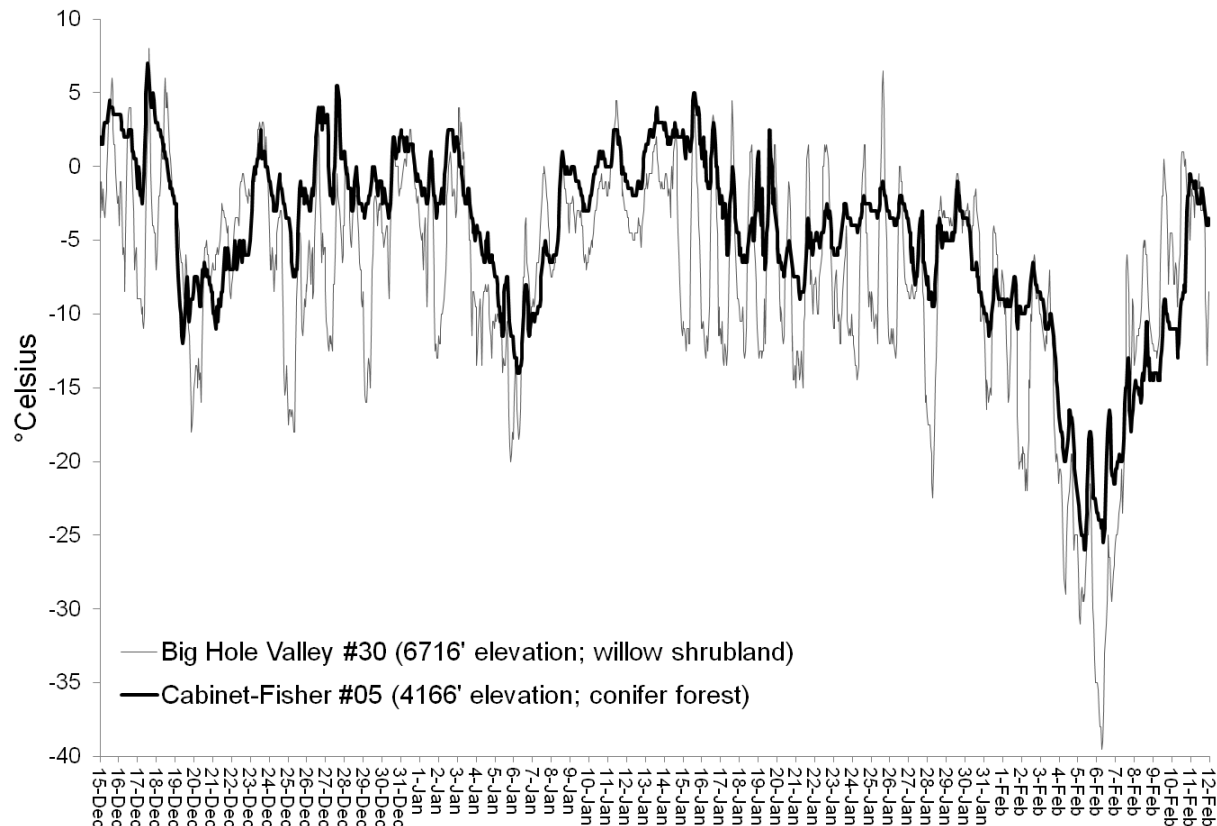
In recent hunting seasons we have recruited moose hunters as collectors of data and samples across the statewide moose population. This has yielded valuable blood samples, liver evaluations, and heads for assessment of chronic wasting disease and *Elaeophora* spp. arterial worm specimens (Table 7).

**Table 7.** *Parasite and disease screening results from moose blood and head samples collected primarily by hunters during 3 of the past 4 seasons in addition to other opportunistic samples.*

	<u>2009-13</u>		<u>2013-14</u>	
	% Positive	N	% Positive	N
Arterial worm ( <i>Elaeophora schneideri</i> )	29%	116	35%	20
Chronic wasting disease	0	110	0	10
Evidence of extensive liver damage	4%	52	7%	14
<b>Serology</b>				
Anaplasmosis (cf)	45%	53	51%	61
<i>Brucella abortus</i>	0	220	0	62
Bovine respiratory syncytial virus	0	219	-	-
Bovine viral diarrhea I	0	220	-	-
Bovine viral diarrhea II	0	220	-	-
Eastern equine encephalitis	0	55	2%	61
Epizootic hemorrhagic disease	1%	169	0	61
Infectious bovine rhinotracheitis	0	219	-	-
Leptospirosis ( <i>L. canicola</i> )	6%	53	3%	61
Leptospirosis ( <i>L. ictero</i> )	9%	54	3%	61
Leptospirosis ( <i>L. grippo</i> )	7%	54	7%	61
Leptospirosis ( <i>L. pomona</i> )	5%	55	0	61
Leptospirosis ( <i>L. hardjo</i> )	2%	54	0	61
Parainfluenza-3	43%	220	-	-

## 2.6. Temperature and snow conditions

Climate and weather conditions can directly and indirectly influence moose populations (Karns 2007, Van Ballenberghe and Ballard 2007). Climatic patterns determining the timing of spring green up, summer precipitation and winter snow conditions can influence survival and recruitment indirectly through effects on forage availability and quality (Van Ballenberghe and Ballard 2007, Brown 2011). Direct effects of climate on moose can be seen in their metabolic response to temperatures (Renecker and Hudson 1986) and the energetic costs of traveling through deep snow.



**Figure 26.** Hourly ambient temperature estimates from 2 example temperature sensors deployed within typical moose winter habitat of the Big Hole and Cabinet-Fisher study areas.

In November 2013, we began monitoring spatio-temporal variation in ambient temperature using field-deployed temperature data loggers (Thermochrom ibuttons, DS1921G-F5; Dallas Maxim Corporation, Dallas, Texas) in each study area (Figure 26). Thermo-loggers were housed within custom radiation shields following Holden et al. (2013) and placed on North side of tree/shrub at 2 m height (Holden et al. 2011). In January, 2013 we also began monitoring snow conditions at moose telemetry locations to document snow depth, snow conditions, and moose sinking depth. Data from these data-loggers and field measurements will be used to validate GIS models developed by the University of Montana Climate Office (Holden et al. 2011) and National Operational Hydrologic Remote Sensing Center (Brennan et al. 2013), respectively. Calibrated model estimates will be used to test the potential effects of climactic factors on moose vital rates.

## Deliverables

Below we list project deliverables (publications, reports, presentations, media communications, and value-added collaborations) stemming from this moose research project, during FY14 (July 2013–June 2014). In addition to those communications listed below, are regular discussions with moose hunters statewide

1. This Annual Report, dated 1 September, 2014, details preliminary results of this multi-year research program

2. Presentations by N. DeCesare and J. Newby, (July 2013–June 2014):

Organization	Location	Date
Swan Ecosystem Center Campfire Program	Holland Lake, MT	July 2013
WCS Community Speaker Series	Laurin, MT	August, 2013
Big Hole Watershed Committee	Divide, MT	August, 2013
Flathead Wildlife Incorporated	Kalispell, MT	October, 2013
MFWP R1, Regional Citizens Advisory Council	Kalispell, MT	December, 2013
MFWP R2, Regional Meeting	Missoula, MT	December, 2013
MFWP R1, Biologists' Meeting	Kalispell, MT	January, 2014
Rocky Mountain Front Land Managers Forum	Choteau, MT	January, 2014
MFWP Bow Hunter Education Workshop	Kalispell, MT	March, 2014
Plum Creek Timber Annual Contractors Meeting	Kalispell, MT	April, 2014
FWP, Wildlife Division Meeting	Fairmont, MT	May, 2014

3. Media Communications by N. DeCesare & J. Newby (July 2013–June 2014):

Group	Media	Date
New York Times interview: Moose declines	Newspaper	October, 2013
Summit Daily (CO) interview: Moose research	Newspaper	October, 2013
The Monocle Daily (London) interview: Moose declines	Radio	October, 2013
Radio New Zealand interview: Moose declines	Radio	October, 2013
Helena Independent Record interview: Moose research	Newspaper	November, 2013
NWF Teleconference interview: Climate & big game	Newspaper	November, 2013
UM Science Source interview: Moose research	Newspaper	February, 2014
Flathead Beacon interview: Moose research	Newspaper	March, 2014
The Nature Conservancy interview: Moose declines	Magazine	April, 2014
KPAX interview: Moose-human conflict	Television	April, 2014
MFWP Outdoor Report: Moose project update	Television	May, 2014
Missoulian interview: Urban moose sighting	Newspaper	July, 2014

#### 4. Other project-related collaborations:

Partners	Title	Activities during FY14
Rick Gerhold & Caroline Grunenwald, University of Tennessee	Development of a serological assay for <i>Elaeophora schneideri</i> detection and surveillance in cervids	*Submitted an approved proposal for funding from MFWP *Providing MT blood samples for lab work
Biologists from western states and provinces (AB, BC, CO, ID, MT, OR, SK, UT, WA, WY)	Assessing range-wide genetic differentiation and spatial distribution of a moose subspecies, <i>Alces alces shirasi</i>	*Submitted an approved proposal for sanction by MFWP *Funding proposals to other agencies and organizations *Coordinating sample compilation across jurisdictions
Ky Koitzsch, K2 Consulting, LLC	Estimating population demographics of moose in northern Yellowstone National Park using non-invasive methods	*Submitted an approved proposal for sanction by MFWP *Providing MT scat samples for fecal pellet morphometry

## Acknowledgements

We are particularly thankful to many private landowners and area residents who graciously allowed us to conduct captures and ground telemetry monitoring on their properties. They also have provided much logistical support (including a friendly rescue after Nick's snowmobile ran out of gas in the Big Hole) and local knowledge about moose distributions and local flora. We are very grateful for the privilege to work on these properties and for all the help.

This project is a large collaboration among many FWP biologists. These include but are not limited to Justin Gude, Jennifer Ramsey, Neil Anderson, Keri Carson, Kevin Podruzny, Keri Wash, Jim Williams, Howard Burt, Graham Taylor, Tonya Chilton-Radant, Kent Laudon, John Vore, Vanna Boccadori, Brent Lonner, Gary Olson, and Ryan Rauscher. Undoubtedly this list should be larger to fully incorporate the many biologists and other personnel who have assisted with coordination of hunter sample collection, harvest statistics, opportunistic sampling of other moose throughout the state. We also acknowledge a great deal of help from other cooperating biologists and agency personnel including Nathan Birkeland, Dave Hanna, Dan Carney, Lorin Hicks, and Allison Kolbe.

Many thanks go to the pilots who have safely conducted capture and telemetry work thus far, including Rick Geiger, Ken Justus, Blake Malo, Jim-Bob Pierce, Joe Rahn, Rick Swisher, Guy Terrill, and Trever Throop (Figure 23).

Funding for this project has been exclusively derived from FWP moose license auction sales, matched with USFWS Grants-in-Aid funds. Many thanks also go to Plum Creek Timber Company, which has supported the project with cooperating helicopter flights during capture and calf monitoring efforts.



**Figure 27.** *MFWP pilot Trever Throop drops into the Big Hole valley for an early morning telemetry and calf-survival flight, Wisdom, Montana, July 2014.*

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